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RESEARCH ARTICLE

Associations of event-scale flow hydrology with fish richness in urbanizing Canadian watersheds of Lake Ontario

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Abstract

Urbanization is associated with declines in aquatic biodiversity and changes to flow regimes. This empirical research examined high temporal resolution (15 min) hydrologic records and associations with fish species richness in eight river systems in the Toronto region, Canada. The dataset spanned approximately five decades and covered the annual post-freshet period to mid-November. The high-temporal resolution flow records allowed estimation of flow acceleration (a measure of the rate of change in flow) in response to rain events. Maximum rising limb event flow acceleration and skew in instantaneous runoff explained a higher proportion of variation than percent urban land use in empirical models with long-term fish records. Models fit using only the most recent decade of records did not produce the same results, likely indicating that analyses of flow with fish diversity require sufficient range in flow conditions for the statistical signals to be detected. Historic fish data are difficult to obtain and pose analytical challenges due to bias and inconsistent collection methods. Despite the data limitations, the study results point to the need for more research into potential causal factors contributing to negative fish richness in urbanizing watercourses with periods of high flow acceleration.

KEYWORDS

aquatic biodiversity, event-scale hydrology, fish richness, flow acceleration, urbanization

1 | INTRODUCTION

Aquatic biodiversity is negatively associated with urban land use and impervious cover (Klein, 1979; Löfvenhaft, Runborg, & Sjögren-Gulve, 2004; Paul & Meyer, 2001; Schueler, 1994; Stanfield, Gibson, & Borwick, 2006; Stanfield & Kilgour, 2006). Urban development brings multiple concurrent environmental changes (e.g., Adamowski & Prokoph, 2013; Boers, DeGraaf, Feddes, & Ben-Asher, 1986; Brabec, Schulte, & Richards, 2002; Costa, Botta, & Cardille, 2003; Dudgeon et al., 2006; Morin & Benyamini, 1977), including to lotic flow regimes (e.g., Dunne & Leopold, 1978; Hammer, 1972; Trudeau & Richardson, 2015). The term "urban stream syndrome" (Walsh et al., 2005) has been coined to capture the degraded condition of urban watercourses. Characteristics of the syndrome include more frequent high flow events, steeper increases in flows in response to rain events, steeper descending hydrograph limbs, water quality changes, fluvial geomorphological changes, reduced biotic richness, and increased prevalence of tolerant species (Walsh et al., 2005).

Detection of biological trends in riverine systems is challenged by the natural variability inherent in numerous processes and interactions of biotic and abiotic factors on multiple spatial and temporal scales (e.g., Burcher, Valett, & Benfield, 2007; Dudgeon et al., 2006; Giller & Malmqvist, 1998; Vannote, Minshall, Cummins, Sedell, & Cushing, 1980; Wipfli, 2005). Further, assessment of biodiversity is challenging because there are numerous potential end points, comprehensive datasets can be difficult to locate, and data collection protocols vary. Fish are common freshwater bioindicators but can be labor-intensive to sample, and differing collection methods introduces sampling bias (Grabarkiewicz & Davis, 2008). In the province of Ontario, Canada, the Ministry of Natural Resources[1] recognized the problem of incompatible data collection protocols for fish. The province of Ontario's systematic sampling methodologies for wadable stream biota were introduced around the year 2000 (Stanfield, 2003) and are documented as the Ontario Stream Assessment Protocol (OSAP; Stanfield, 2013).

Despite numerous sources of uncertainty, negative aquatic biotic responses to impervious cover at about 10% have been identified in

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[1]Ontario's Ministry of Natural Resources was renamed the Ministry of Natural Resources and Forestry in 2014. The former name is retained because references and communications cited date prior to the name change.

very different locations globally (Schueler, Fraley-McNeal, & Cappiella, 2009), including the Ontario Great Lakes Region (e.g., Stanfield & Kilgour, 2006), Washington State (e.g., Chin, 2006), and Australia (e.g., Walsh, Sharpe, Breen, & Sonneman, 2001). This fairly consistent response to increased impervious cover supports the hypothesis that hydrological changes are one cause of biodiversity decline.

Changes in hydrologic regime of relevance for biota are poorly understood because few studies have been undertaken at the appropriate temporal scales for flows. Due to data availability, hydrologic analyses are typically undertaken on coarse temporal scales (i.e., daily, weekly, or monthly) (e.g., Clausen & Biggs, 1997; Monk, Peters, & Baird, 2012). Correlations of aquatic biotic community response to changes in flow frequency, magnitude, and duration (e.g., Poff & Zimmerman, 2010) have been identified using daily and monthly temporal scales for flow data. Causal factors are difficult to identify; Burcher et al. (2007) identified negative influences on fish density associated with cascading geomorphic and erosional factors propagated from urban land use change.

Few studies have assessed the relationship of hydrologic changes at the rain event scale with urban land use and biota. With high temporal resolution flow data, the rising hydrograph limb, which rises in response to rain events, can be quantified as a rate of change in flow, or flow acceleration. With appropriate data, flow acceleration can be estimated as the change in flow per second, that is, in units of cubic meters per second. This measure of flow is not in common use because high resolution flow data are not commonly available. Using Environment Canada hydrologic data at 15-min increments, Trudeau and Richardson (2016) identified dramatic changes in event-scale hydrology with urbanization, including increased rising limb event flow accelerations with increasing watershed urbanization. Empirical models within the study area indicated consistently increasing rising limb event flow accelerations up to at least 70% urban land use on a watershed basis (Trudeau and Richardson, 2016).

The 15-min event-scale records are commensurate with changing water column conditions experienced instantaneously by biota and by habitat features affected by high energy water and sediment impulses during rain events. Daily hydrologic records cannot sufficiently reflect instantaneous flow changes, especially in urbanized watersheds with flashy flow responses. Event flow characteristics included in our analyses focused on the rising limb of rain events because it is during this interval that aquatic biota needs to shelter (Tetzlaff, Grottker, & Leibundgut, 2005a), adjust, or otherwise sustain themselves as the effects of a storm move through a watershed. The rising limb, until peak event flow, is also the period of highest energy impulse resulting from rain events. Thus, the rising limb of event hydrographs is relevant to understanding both direct effects of changes in the water column and indirect effects arising as a result of alterations to habitat features of the physical stream channel (e.g., substrate stability changes with increased stream power imparted during rain events).

In this study, we used hydrologic data at 15-min increments for eight river systems confluent with Lake Ontario (Figure 1) in Ontario, Canada, at locations matched with fish observation sites in an intensively urbanizing region. Study watersheds ranged in size from 37 to 806 km^2 and had urban percent (UP) land use from 0.06% to 87.6%. This study was the third phase of research using this high temporal resolution hydrologic dataset in this study area. In the first phase, temporal analyses for two of the watersheds (the Don and Humber Rivers) established that event-scale hydrologic trends were detectable within the study area over four decades (Trudeau & Richardson, 2015). The second phase analyzed changes across spatial scales of watershed size and percent urbanization using 27 watersheds within the study area (Trudeau and Richardson, 2016). The empirical statistical models developed in the second phase quantified the dramatic effect that urbanization has on event-scale flows, including flow acceleration. In the first two phases, watershed characteristics, percent urban land use, and rainfall were independent variables used to empirically model hydrologic responses, including total flow, event-scale flows, and flow acceleration. In this third phase, event-scale hydrologic characteristics, including rising limb flows, rising limb flow accelerations, and rising limb stream power, were then used as independent variables to model fish richness, along with watershed characteristics and other independent variables as described herein.
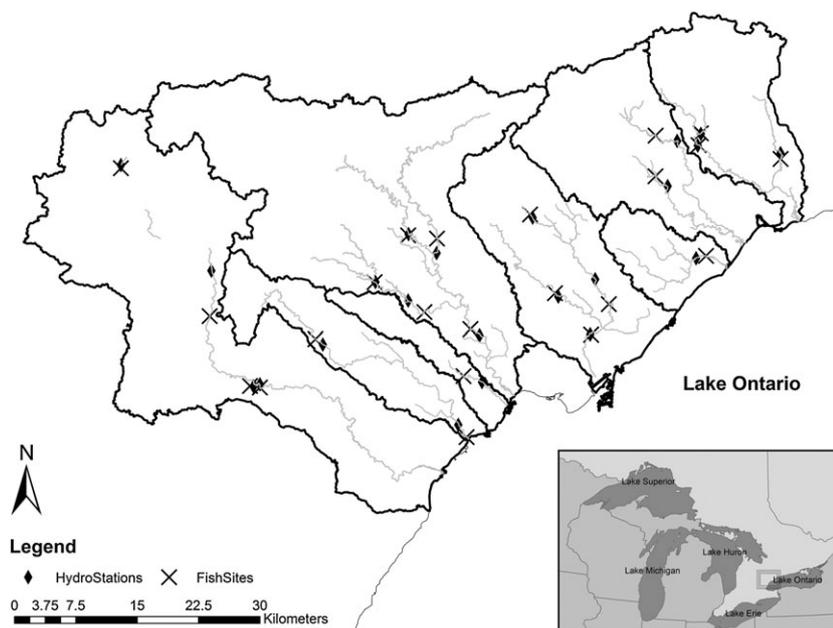
FIGURE 1 Study area map. The study area is on the north shore of Lake Ontario in the Greater Toronto Region, Ontario, Canada. The eight study river systems, from west to east by confluence with Lake Ontario, were Credit River, Etobicoke Creek, Mimico Creek, Humber River, Don River, Highland Creek, Rouge River, and Duffins Creek (solid black lines are river system boundaries; light grey lines are river networks). Fish sites are indicated by crosses and paired Water Survey of Canada hydrologic stations by diamonds

The Toronto region has undergone intensive urbanization during the study period, making the city of Toronto now the fourth largest city by population in North America (City of Toronto, 2014). The historic urban core of the study region is serviced by a combined sewer system, but the study watersheds were predominantly serviced by conventional separated sewer systems over the timeframe of available hydrologic data (1969 to 2010). In the most recent years, there has been some evolution of stormwater management practices to include low impact development stormwater measures in some satellite community subdivisions; runoff quantity control was only introduced in Ontario in the 1980s and low impact development post-year 2000 (CVCA and TRCA, 2010).

Changes in the fish community within the study area have been documented as early as the 19th century, and a first management intervention was introduced in 1807 to preserve salmon stocks (City of Toronto, 2012). The common carp was introduced to Lake Ontario in the 1870s and other nonnative fish followed, either by introduction or invasion, including brown trout in 1929, white perch by 1948, and round goby in 1998 (City of Toronto, 2012). A database with fish records from the early-20th century to 2010 was compiled through collaboration with the Credit Valley Conservation Authority (CVCA), the Toronto and Region Conservation Authority (TRCA), Ontario government agencies, and individual researchers.

It was expected that UP would be a better predictor of fish species richness than any of the hydrologic characteristics. Correlative studies in urban areas have identified negative associations of water chemistry and thermal regime change with biota (e.g., Chu & Jones, 2011; Wallace, Croft-White, & Moryk, 2013). In urbanizing watersheds, it was assumed that changes in water chemistry and thermal regime occur concurrently with changes in hydrologic regime. For this reason, the statistical signal from urban extent was expected to integrate changes associated with all urban effects, including hydrologic, if detectable.

In this exploratory empirical analysis, we asked what characteristics of urbanizing watersheds and hydrologic characteristics explain the highest proportion of variation in fish species richness. We also examined biases and issues in the fish database to inform the question whether recent data (i.e., space in current time) can substitute for long-term fish records in this type of study.

2 | MATERIALS AND METHODS

2.1 | Database

Data for this study were obtained from multiple sources:

- Environment Canada Water Survey of Canada hydrologic data with 15-min temporal resolution from 1969 to 2010;

- Watershed scale urban land use delineation estimated by year, with scattered temporal coverage between 1969 and 2011 (multiple sources, described in Trudeau and Richardson, 2016);

- Baseflow index estimates (Dr. J Buttle pers.comm. (2015) and Neff, Day, Piggott, & Fuller, 2005)

- Fish species characterization, including invasive species (R. Eakins, pers. comm., 2013)

- Fish data for eight river systems at a temporal resolution of day or month, between 1928 and 2011. Recent fish data (i.e., since 2000) collected by the CVCA, the TRCA, and the Ontario Ministry of Natural Resources (OMNR) were accessed through data-sharing agreements for the Provincial database, Flowing Waters Information System. The Royal Ontario Museum provided historic fish records. Dr. Gordon Wichert, SLR Consulting (Canada) Ltd, provided an electronic version of his PhD (1995) database. Published reports and theses were also mined for data (Wainio, 1959; Johnson and Owen, 1966; Wainio and Hester, 1973; Steedman, 1984; Martin-Downs, 1988; Wichert, 1995).

Fish were the only biotic community with sufficient temporal and spatial data for analyses. The database included all recorded fish presence observations in the available records, including native, nonnative, and invasive species. Fish species identification followed OMNR Fish Codes (Stirling, 1999). The coding system identifies a group code in addition to species codes within the group. For instance, *Cyprinidae*—carps and minnows—is Code 180, and there are 49 species within this group, identified with Codes 181 to 229 (Stirling, 1999). No editing of species codes or fish presence records was performed during database compilation; data were used per the original source material records.

All hydrologic data used were during the post-freshet year (May 26th to November 15th) to reduce statistical noise introduced by variable winter melt conditions that occur within the study area. Preparation of the hydrologic data, matched to hourly rainfall data, was reported in Trudeau and Richardson (2015). Development of UP estimates, using historical aerial photographs, was reported in Trudeau and Richardson (2016). Estimated UP by watershed was the ratio of the area of urban extent upstream of each hydrologic station relative to the watershed area, expressed as percent. Watershed area, channel slope, and basin slope upstream of fish observation sites were estimated using the methodology described in Trudeau and Richardson (2016) for hydrologic stations.

Flow acceleration (in units of cubic meters per second per second) was calculated for rising limb event flows based on the difference between each sequential 15-min flow record, divided by 15 min (converted to seconds). Custom R scripts were used to isolate events in the flow records and to calculate accelerations (see Trudeau and Richardson (2016) for additional detail). The maximum acceleration used in empirical statistical models was the highest calculated rising limb acceleration within the flow record for each hydrologic site for each year within the database.

Stream power was also included in the database for its potential as an indicator of habitat alteration. An event stream power measure was estimated per unit length of stream, normalized for watershed basin area. Stream power per unit length of stream (Gordon, McMahon, Finlayson, Gippel, & Nathan, 2004) is expressed as (Equation 1).

$$\Omega = \rho g Q S \qquad (1)$$

Where:

Ω is stream power (Watts per unit length of stream, W/m)

ρ is the density of the liquid (kg/m^3)

Q   is flow, m³/s; in this case, rising limb event flows at 15-min increments

g   is acceleration due to gravity, m/s²

S   is channel slope, used as an approximation of energy slope, dimensionless

Larger streams have larger event flows, so the power per unit length estimate was divided by the area of each watershed for comparison across the study fish basins. Note that channel sinuosity was not estimated or included in the calculation; the power measure was intended to be exploratory with respect to change in unit stream power with change in event flows across heterogeneous watersheds.

For some older fish observation sites, locations were approximated from paper map records. Sites from various available sources were grouped based on proximity to each other within the same stretch of watercourse with no intervening tributary confluences (called 'grouped sites'); sites within a grouped site were up to approximately 1,600 m of river length from the database site location.

The year of fish data collection and the year of air photos used to estimate urban extent often did not match. Temporal matching of fish to urban observations was made according to the following rules:

1. For UP estimates less than 0.5% in 1969, records as early as 1949 were matched to 1969 because there was virtually no urban development in the watershed.

2. For UP estimates over 0.5% but less than 3%, fish records up to 10 years earlier were matched because very little urban development had taken place during the earlier decade (e.g., some 1959 fish records were matched to 1969 land use estimates).

3. For other UP estimates, fish records were matched within 5 years of the available UP estimate; most fish records were matched to UP estimates within 3 years of the fish record.

The resulting database had 22 sites with 49 records in eight river systems (Table 1). Appendix A Figures A1, A2, and A3 identify the watersheds upstream of the fish observation sites and the hydrologic site locations; mapped site numbers correspond to fish observation sites identified in Table 1.

2.2 | Independent variables

Independent variables used to model fish richness (the dependent variable) can be characterized in four groups: (a) location of fish observation site relative to the mouth of the river system, (b) watershed characteristics upstream of the fish observation site (area, slopes, total stream length, and baseflow), (c) hydrologic characteristics (power [see Equation 1], runoff and acceleration), and (d) land use (urban percent cover on a watershed basis). Table 2 describes variables used in statistical analyses. *Year* is the year of fish observation. Note that not all results for independent variables fit are reported in detail herein.

TABLE 1 Fish observation sites and watershed information. A summary of the sites available in the database, numbered to identify location on watershed maps

Map Site number	River system	Drainage area (km²)	Strahler stream order at fish observation Site	Number of years with UP records	Values of UP in the record (%)
1	Credit	636	6	3	11; 13; 14
2	Credit	439	6	2	9; 13
3	Credit	131	5	3	15; 17; 24
4	Credit	60	4	2	23; 24
5	Etobicoke	219	5	3	21; 48; 57
6	Etobicoke	78	5	1	12
7	Mimico	63	4	2	56; 88
8	Humber	806	7	2	4; 9
9	Humber	297	7	1	1
10	Humber	190	6	2	2; 7
11	Humber	188	6	1	0
12	Humber	143	6	1	0
13	Don	312	5	3	57; 66; 72
14	Don	141	4	3	31; 49; 62
15	Don	83	4	4	26; 40; 51; 66
16	Don	37	4	1	37
17	Highland	93	4	4	67; 72; 83; 85
18	Rouge	183	6	4	4; 6; 36; 39
19	Rouge	71	5	4	1; 1; 2; 3
20	Duffins	108	6	1	1
21	Duffins	88	6	1	0
22	Duffins	62	5	1	1
	22 Sites in 8 River Systems			49 records	

TABLE 2 Variables included in statistical analyses. A summary of potential explanatory variables and the dependent variable used in the analyses

Explanatory variable	Description	Units	Notes
Location			
Distance from mouth	Euclidean distance of fish site from mouth of Lake Ontario	m	Tested in GLMM for long-term records
Watershed characteristics			
Drainage area	Watershed area upstream of the fish sampling site	m^2 or km^2, depending on scale of analyses and discussion	Tested in GLMM for long-term records
Channel slope	Slope of the watercourse upstream of a fish observation site	$m \cdot m^{-1}$	
Basin slope	Slope of the watershed basin upstream of a fish observation site	$m \cdot m^{-1}$	Tested in GLMM for long-term records
Total length of stream upstream of fish sampling site	Estimated from total length of upstream tributaries	m	Tested in GLMM for long-term records
Baseflow index (BFI)	Ratio of baseflow to streamflow	unitless	
Hydrologic variables (applicable to May 26th to November 15th in all cases)			
10th percentile event stream power per unit length of stream per square meter watershed area	Watts per channel length ($kg\ m^2 \cdot s^{-3}$ per m) divided by drainage basin area (m^2). Watts calculated as the product of water density ($1000\ kg \cdot m^{-3}$), gravity acceleration ($9.8\ m \cdot s^{-2}$), rising limb event flow ($m^3 \cdot s^{-1}$), channel slope ($m \cdot m^{-1}$).	Watts per channel unit length (m) per m^2	10th percentile calculated on a yearly basis using all available events; Q equals the 15-min increment rising limb event flows
90th percentile event stream power per unit length of stream per square meter watershed area	Watts per channel length divided by drainage basin area (m^2).	Watts per channel unit length (m) per m^2	90th percentile calculated on a yearly basis using all available events; Q equals the 15-min increment rising limb event flows
Mean flow per square meter drainage area ($MF \cdot m^2$); also referred to as mean instantaneous runoff	Mean flow ($m^3 \cdot s^{-1}$) divided by basin area (m^2)	$m \cdot s^{-1}$	Mean flow calculated on a yearly basis using as Q the 15-min increment flows, including event and nonevent flows
			Included event and nonevent instantaneous flows
Skew of $MF \cdot m^2$ (called skew in runoff)	Skew of the mean instantaneous runoff	unitless	Calculated on a yearly basis including all flows for the year (event and nonevent). Tested in GLMM for long-term records
Mean rising limb event acceleration	Mean flow acceleration ($m^3 \cdot s^{-2}$) from one 15-min increment to the next during the rising limb of event flows, up to peak event flow	$m^3 \cdot s^{-2}$	Calculated on a yearly basis for event acceleration >0 $m^3 \cdot s^{-1}$ during the rising limb of events up to peak flow. Tested in GLMM for long-term records
Maximum event acceleration	Maximum flow acceleration ($m^3 \cdot s^{-2}$) during event flows	$m^3 \cdot s^{-2}$	Maximum event acceleration in a year. Tested in GLMM for long-term records
Skew of rising limb event acceleration	Skew of the mean rising limb event acceleration	unitless	Skew calculated on a yearly basis including the mean rising limb acceleration of all recorded events
Land use			
Urban percent (UP)	Percent of the watershed upstream of the hydrologic gauge that is urban land use	unitless	Tested in GLMM for long-term records

Dependent Variable	Description	Units	Notes
Fish species Richness	Number of fish species per site; species were defined by OMNR species codes (Stirling, 1999)	Count	Some species codes are generic for a group rather than an individual species. See Stirling, 1999.

Note. GLMM = generalized linear mixed models; OMNR = Ontario Ministry of Natural Resources.

2.3 | Statistical analyses

The analyses of fish records could only include fish richness as measured by species present. Although some data sources quantified the numbers of individual fish by species, inconsistent data collection approaches (see Section 4) precluded analyses of abundance. A database of all fish records at the study sites for all available years with fish records post-1940 had 129 species (see Appendix B). The analyses required concurrent estimates of urban percent land use and hydrologic flows, as well as fish records; in the full dataset analyzed, there were 115 species because some years with fish records could not be used.

The entire set and two subsets of fish data were analyzed. The two subsets were (a) only records for fish species present in the earliest comprehensive surveys (1940s and 1950s; 55 species) (called the *long-term record database*) and (b) all fish species records from 2000 to 2010 (called the *recent database*; 99 species). The full dataset and the long-term database included 49 records at 22 sites identified in Table 1. The recent dataset included 17 records at 11 sites; all sites in the recent dataset were part of the other two datasets.

The two subsets of data were used to eliminate some of the temporal bias in the dataset. By including only species with a long-term record (i.e., subset 1, the long-term record database), the presence of these fish across sites and through time was assumed to be comparable, except where effort bias potentially affected the number of species identified (see Section 4). Since about year 2000, fish surveys in the study region were undertaken using OSAP or compatible protocols. By including only records subsequent to 1999 (i.e., subset 2, the recent database), the database potentially excluded sampling methodology bias. The full dataset was analyzed to illustrate temporal bias. The recent dataset was further analyzed using only records with fish identified to the species level to assess whether inclusion of records with only family or genus altered the results.

There were numerous multi-collinear independent variables and relatively few degrees of freedom due to the limited number of hydrologic sites matching UP estimates and fish sites. To minimize multicollinearity issues, the analysis was undertaken in two phases: (a) independent variable selection and (b) fish richness model fitting using the selected independent variables. Some watersheds had more than one record through time, raising the possibility of non-independent observations. Consequently, statistical models fitted to the data included random terms to account for this non-independence. Some watersheds were nested within larger catchments, for example, upstream of a branch confluence with mainstem channels or within the upper to mid-watershed mainstem regions (see Appendix A). At the event-scale, flows at downstream mainstem sites were assumed to be independent of the upper catchment flow sites due to very different catchment sizes, the additional flows from other contributing catchments, and the resulting differences in flow magnitudes.

The goal of the first phase of the analysis was to identify independent variables that explained the highest proportion of variance for (a) site or watershed characteristic and (b) hydrology. As a preliminary step, the interrelationships of variables were assessed with principal component analysis (PCA) using the R package 'vegan' (Oksanen et al., 2013). PCA analyses included fitting a fish richness vector to the PCA plots of the first two components. Predicted species richness was also estimated and plotted. A permutation test was used to assess significance of the association between fish richness and the PCA scores.

The goal of the second phase of statistical analysis was to fit models using the selected independent variables to predict fish richness. Two types of models were fit: (a) generalized linear mixed models (GLMM) with random intercepts by site and Poisson distribution using two independent variables (one physiographical and one hydrologic or UP) and (b) fixed effects linear models with Poisson distribution using up to four independent variables.

Generalized linear mixed models were fit using the R package 'lme4' (Bates, Maechler, Bolker, & Walker, 2015). Response, Pearson, and deviance residuals of the best GLMMs were plotted and visually inspected for structure using R code developed by Zuur, Ieno, Walker, Saveliev, and Smith (2009). GLMM estimates for *p*-values and Akaike's information criterion are approximate (Zuur et al., 2009). Over-dispersion in the residuals was assessed by the ratio of chi-square to model degrees of freedom using publicly posted R code (http://glmm. wikidot.com/faq). Where there is over dispersion in GLMMs, Bates et al. (2015) recommend against a quasi-Poisson distribution for GLMMs. Where GLMM models had over-dispersion in the residuals, a negative binomial model (with fixed effects only) was fit for comparison using the R package 'MASS' (Ripley, 2015).

Linear models, with fixed effects only, were fit using the 'glm' command (R Core Team, 2012) for up to four variables in combination, representing time, physiography, hydrology, and UP. Models were compared based on Akaike's information criterion, *F*-tests using the 'analysis of variance' command (R Core Team, 2012) and a calculated residual deviance relative to the null (1-[residual deviance/null deviance]), reported as *Pseudo R^2*. The 'analysis of variance' command in the R Package 'car' (Fox and Weisberg, 2011) was used to assess the partial R^2 of fixed effects model variables; Type II where there were no interactions in the model and Type III where there were interactions.

3 | RESULTS

Following a section on overall trends in the database, the results are organized by database analyzed. The long-term database subset results are presented first because these provide a basis for comparison with the full database (presented second) and the recent database (presented third).

3.1 | Overview of trends in the database

Research uncovered 2,700 records for fish at 497 sites between 1927 and 2010. However, the study database had only 49 records at 22 sites because of limited matching urban-hydrologic data available. Within this study database, there were 60 more fish species recorded in 2011 compared with the 1940s and 1950s fish surveys. Only seven of the 60 were introduced species not included in the long-term species record (R. Eakins database, pers.comm., 2013). Sixteen additional species were in the *Cyprinidae*—carps and minnows group (see Stirling,
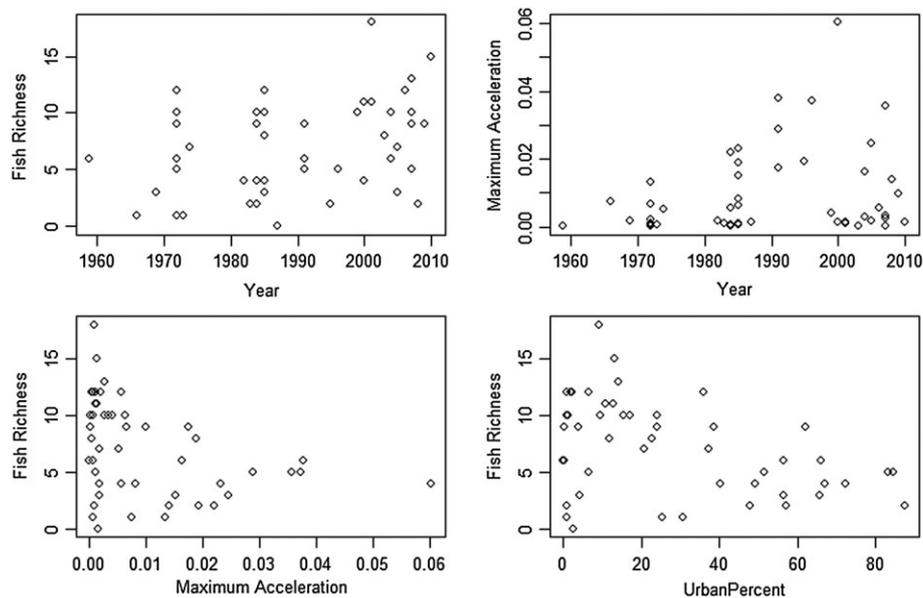
FIGURE 2 Raw data plots. Fish species richness, using the long-term record database, by year (upper left, Spearman rho = 0.28, *p* = 0.05), by maximum acceleration (lower left, Spearman rho = −0.39, *p* = 0.005) and by urban percent (lower right, Spearman rho = −0.37, *p* = 0.009); maximum acceleration ($m^3 \cdot s^{-2}$) is also plotted by year (upper right, Spearman rho = 0.26, *p* = 0.07)

1999). Some of the additional species codes may indicate misclassification error (or introduced species) rather than bias (e.g., coho salmon); others include more recent records of fish identified only at the level of family or genus and hybrid species not identified in earlier studies.

3.2 | Long-term record: relationship of fish species and hydrologic characteristics

Fish richness in the long-term database ranged from 0 to 18 species per site. All species codes in the long-term database identified fish to the species level with one exception, identified to genus level (S177 *Catostomidae–Moxostoma sp.*). Raw data plots indicated that there was potential positive bias in fish richness with time; the highest fish richness records per site were recorded in more recent years, potentially indicating effort bias (Figure 2, top left panel). The highest maximum acceleration records occurred during the most recent years

(Figure 2, top right panel), consistent with high correlation with increased UP (Spearman correlation = 0.82, *p* = 5.3e-13), which also increased over the time period. The highest fish richness records occurred at sites that had the lowest maximum accelerations (Figure 2, lower left panel). The highest fish richness records also occurred at lower UP, but the highest fish richness did not occur at the very lowest UP (Figure 2, lower right panel).

Increased fish richness with time was observed at many sites (Figure 3, left panel), likely indicating effort and/or methodological bias in the dataset. Many individual observation sites recorded increased fish richness with increased UP and increased fish richness with maximum acceleration (Figure 3, middle and right panels, respectively). Increased event acceleration is positively associated with UP (Trudeau and Richardson, 2016).

Using all potential independent variables (see Table 2), the first two principal components explained 59.6% of the variance (37.2%
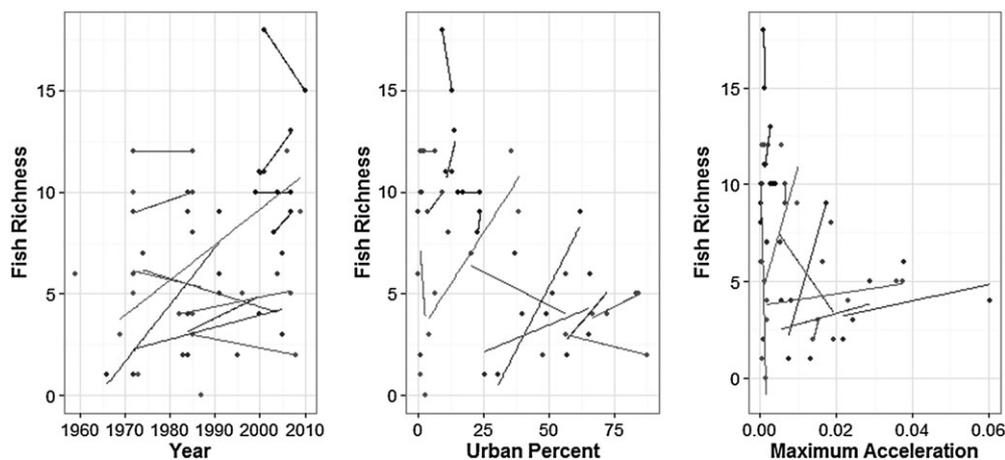
FIGURE 3 Raw data plots with trends by site. Raw data plot of fish richness, using the long-term record database, with year, urban percent, and maximum acceleration ($m^3 \cdot s^{-2}$) by site. Linear trend lines by site are superimposed
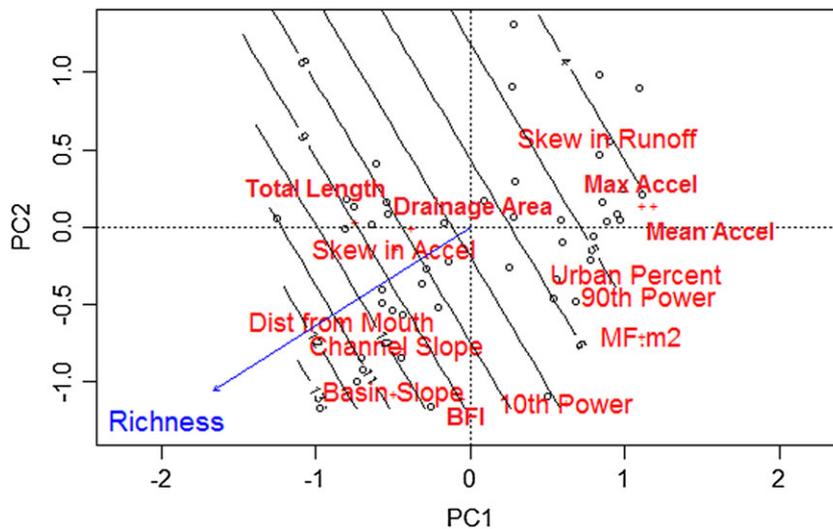
FIGURE 4 Principal Components Analysis (PCA) plot for long-term database. PCA results for fish monitoring sites (black circles) and independent variables (text) with species richness vector (lower left quadrant) for the long-term database. Predicted species richness, estimated from PCA scores, is overlain (black diagonal lines with predicted number of species)

and 22.4%, respectively; Figure 4). *Distance from mouth* (−0.83) and *total length* (−0.75) had the most negative loadings on PC1, whereas the acceleration variables, *maximum acceleration* (1.11) and *mean acceleration* (1.19), and *90th percentile power* (1.16), had the most positive loadings on PC1. PC1 thus reflected the negative relationship of event acceleration and highest power with the length of stream network and distance of watershed from Lake Ontario. *Basin slope* (−1.07), *10th percentile event power* (−1.13), *BFI* (−1.11) had the most negative loadings on PC2, and *skew in runoff* (0.58) had the most positive loading on the second component. This component reflected characteristics of watersheds in the study region (e.g., basin slope and BFI) that are negatively associated with skew in runoff (which generally increases with increasing urban cover). The two event power variables (10th and 90th percentiles) were in the same quadrant of the PCA, indicating other variables more strongly influence the span of these two components. The fish richness vector was most highly correlated with PC1 (−0.84) (PC2 correlation = −0.54); vector R^2 = 0.37 (*p*-value = 0.001).

Eight independent variables were identified to model fish richness, using the process described in Section 2 to reduce the number from 14 potential variables (see Table 2). Four of the eight variables pertained to physiographic characteristics of the watersheds and fish site locations (*total length*, *distance from mouth*, *drainage area*, and *basin slope*). The remaining four pertained to hydrologic characteristics (maximum acceleration, mean acceleration, skew in runoff, and UP). To model the bias in the database, Year was also tested as an independent variable.

The four physiographical variables were fit individually in GLMMs with random intercepts by site to predict fish richness. *Total length* of stream explained the largest proportion of variation in fish richness for this initial step and had the advantage that it was not highly correlated with any of the four additional variables fit in the next step of model development (i.e., three hydrologic variables and UP).

Generalized linear mixed models were then fit using total length and a second independent variable representing hydrology or UP, and the results of these four models were compared with each

TABLE 3 Fish richness model comparisons with up to four independent variables for the long-term database

Model	Independent variables	Coefficient ± SE	*p*-value	Residual deviance (RD)	AIC	Pseudo R^2
Null				Null deviance (ND): 129.0 on 48 df		
A. Time and physiography (1,2)	Log_{10} total length Year	0.74 ± 0.13 0.014 ± 0.004	**3.95e-09** **3.4e-04**	85.1 on 46 df	265.96	0.34
B. Time, physiography, and hydrology (1,2,3)	Log_{10} total length Year Log_{10} maximum acceleration	0.69 ± 0.13 0.018 ± 0.004 −0.36 ± 0.08	**7.5e-08** **4.6e-06** **7.9e-06**	65.1 on 45 df	248.05	0.50
C. Time, physiography, and UP (1,2,4)	Log_{10} total length Year Log_{10} UP	0.62 ± 0.13 0.025 ± 0.005 −0.32 ± 0.088	**2.5e-06** **7.3e-07** **2.5e-04**	72.2 on 45 df	255.07	0.44
D. Time, physiography, hydrology, and UP (1,2,3,4)	Log_{10} total length Year Log_{10} maximum acceleration Log_{10} UP	0.67 ± 0.13 0.019 ± 0.005 −0.33 ± 0.124 −0.042 ± 0.14	**5.0e-07** **4.5e-04** **7.0e-03** 0.763	65.1 on 44 df	249.96	0.50
E. Time, physiography, and interaction of hydrology with UP (1,2,3:4)	Log_{10} total length Year Log_{10} maximum acceleration: Log_{10} UP	0.716 ± 0.128 0.019 ± 0.005 0.060 ± 0.037	**2.4e-08** **1.7e-04** 0.100	82.5 on 45 df	265.36	0.36

Note. The dependent variable for all models was fish richness. Bold entries indicate significant *p*-values. InterceptsZ are not provided for simplicity.

corresponding model that used only total length. The addition of maximum acceleration was a significant improvement (5.1% additional deviance explained; χ^2 = 4.96, p = 0.026). Skew in runoff was also an improvement (5.0% additional deviance explained; χ^2 = 4.81, p = 0.028) although the over-dispersion reduces the reliability of this estimate. Mean acceleration was not a clear improvement because it barely passed the statistical significance threshold of 95% confidence (3.8% additional deviance explained; χ^2 = 3.68, p = 0.055). The addition of UP did not statistically improve the total length model (0% additional deviance explained; χ^2 = 4e-04, p = 0.98).

Results for the analyses using up to four independent variables were consistent with those for the GLMM analysis (Table 3); richness was predicted using combinations of up to four groups of variables (brackets indicate the independent variable used): (a) time (Year), (b) physiography (total length upstream of the fish site, log_{10} transformed), (c) hydrologic (maximum event acceleration, log_{10} transformed), and (d) UP (log_{10} transformed). Model B (time, physiography, and hydrology) explained 5.5% additional deviance compared with Model C (time, physiography, and UP). The results for all models indicated a positive effect of Year.

An analysis of deviance to assess the partial R^2 of the independent variables indicated the role of Year in explaining variance within the models. In Model B (with maximum acceleration), Year (LR χ^2 = 21.2, p = 4.1e-06) explained less variation than Year (LR χ^2 = 25.5, p = 4.4e-07) in Model C (with UP), indicating that maximum acceleration explained some of the variation through time encompassed by Year in Model C. In Model D (year, maximum acceleration, and UP), UP (LR χ^2 = 0.09, p = 0.763) explained virtually none of the variation.

3.3 | Analyses using all fish records

Fish richness in the database with all fish observations ranged from 1 to 18 species per site. The results for GLMM models were similar to those for the long-term database; coefficients were within the range of the standard error for model coefficients fit to the long-term database. However, these GLMM models had over-dispersion in the residuals making the coefficient estimates less reliable. As was the case for the long-term database, the addition of UP did not statistically improve the total length model (0% additional variance, χ^2 = 0.007, p = 0.93).

The results for the four-variable analyses were also similar to those for the long-term database. For instance, in the four-variable model (i.e., the Model D equivalent), UP was not a statistically significant independent variable (χ^2 = 0.3, p = 0.561), Year had a statistically significant positive effect (χ^2 = 13.5 p = 0.0002), and maximum acceleration had a statistically significant negative effect (χ^2 = 5.8, p = 0.016).

3.4 | Analyses using the recent database

Sites in the recent database (2000 to 2010) had between two and 18 species each. Although the number of species was higher overall, there were six species codes missing from the recent database relative to the long-term database: *Moxostoma sp.* (S177); eastern silvery minnow (S190); channel catfish (S234); tadpole madtom (S236); *Noturus sp.* (S242); banded killifish (S261; see Appendix B). Note that these species

may have been recorded at other locations within the timeframe of the recent database.

The range of hydrologic variables was reduced by half in some cases compared with the long-term database (see Appendix C). Further, minimum values in the recent database for maximum acceleration ($m^3 \cdot s^{-2}$) and mean runoff ($m^3 \cdot s^{-1}$) were 2 times higher than those in the long-term database.

Results for the GLMMs using this database differed from the long-term database results for three of the additional variables to total length. UP was a statistically significant improvement (χ^2 = 7.30, p = 0.007) with a negative coefficient. Mean acceleration was a clear improvement (χ^2 = 6.28, p = 0.012), but skew in runoff was not an improvement (χ^2 = 2.90, p = 0.089). However, the result for the addition of maximum acceleration was similar as it was a significant improvement (χ^2 = 5.80, p = 0.016) to predict fish richness. These results were generally consistent for a subset of recent records identified to the species level.

The analyses using up to four independent variables also differed from those for the long-term database. Year was not statistically significant in any of the models using all recent records and had a statistically significant positive coefficient using a subset of records identified to the species level. Models with three independent variables (excluding Year) had no statistically significant independent variables using all recent records; maximum acceleration and UP each had statistically significant negative coefficients in three-variable models with Year and total length when fit to a subset of records identified to the species level.

4 | DISCUSSION

Despite limitations of the fish records (see Section 4.1 following), the empirical modelling results consistently point to flow acceleration characteristics as hydrologic variables of potential concern with respect to fish community richness. Additions of the hydrologic characteristics (maximum acceleration of the rising limb event flows and skew in runoff) to statistical models, each with total length of stream as a second independent variable, indicated negative associations with fish richness. Increasing positive skew in runoff indicates an increasing right tail of the distribution, which is characteristic of increasing flashiness of urbanizing streams and therefore also indicative of increased rising limb accelerations, among other hydrologic changes. Moreover, maximum acceleration also improved the explained variation in models fit with Year, total length, and UP in combination. Similar results even held for empirical models fit to a database with all fish records, which had additional species in later years (see Appendix B); in other words, even with large positive bias in fish records concurrent with increasing flow acceleration, a signal for negative associations of fish richness with maximum acceleration was still detectable.

Tetzlaff et al. (2005b) had previously examined the relationship of urban land use with flow acceleration but no consistent relationship was identified. The Tetzlaff et al. (2005b) study used two sets of flow records (1-hr temporal resolution for 16 catchments, and 6-min resolution for three catchments) and included both modeled and empirical data in the analysis. Results were inconclusive with respect to the

relationship of urban cover with acceleration (Tetzlaff et al., 2005b) possibly because data were analyzed as separate databases, catchment characteristics (other than size) were not included, and rainfall was not taken into consideration. Trudeau and Richardson's (2016) empirical statistical models to predict flow characteristics included independent variables of total rainfall, watershed size, catchment slope, a baseflow variable, and percent urban land use. A clear increasing relationship of flow acceleration with urban land use was empirically evident in that study.

Surprisingly, the extent of urban land use (UP in the models) did not explain more variation than the hydrologic characteristics (maximum acceleration of the rising limb event flows and skew in runoff), except in an analysis of only the most recent records in isolation (2000 to 2010). The result that UP did not improve the explained variation of the long-term database, whereas hydrologic variables did, was surprising considering the numerous concurrent changes, also negatively associated with fish richness that occur with urbanization. Two possible explanations for this result, which are not mutually exclusive, concern (a) concurrent changes and (b) timing of the effects of flow acceleration alterations.

With respect to the concurrent changes, there may have been additional sampling efforts in watersheds as their urban cover increased, which could have reduced the signal from urban cover (assuming more fish species were identified with the increased effort). In addition, changes in event flow acceleration may have occurred in rural areas as a result of changed agricultural or other practices through time. These changes would have increased the acceleration signal without a coincident change in urban cover. Also, improvements to sanitary sewage treatment were made since the 1940s, with improved fish richness in previously impaired streams (Wichert, 1994). Year served as a surrogate for temporal changes through the 42-year study period and the bias in the database is evident in the statistically significant positive coefficient for Year in the four-variable model analyses. Year explained more variation than maximum acceleration or UP, which each had negative coefficients (although UP was not statistically significant using the long-term database). Given well-documented negative associations of increased urban land use with aquatic biodiversity in the literature, the positive effect of Year on fish richness is most likely explained by changes to fish data collection (see Section 4.1 following) and, potentially, other concurrent changes.

With respect to the second possible explanation, the result that hydrologic variables were more strongly negatively associated with fish richness than UP may reflect very early negative effects of flow regime change on fish richness. Changes in mean acceleration in the study area began to occur at very low percent urban land use, possibly as early as 4% UP (Trudeau and Richardson, 2016). If flow acceleration has a strong negative effect on fish richness, the effects may occur prior to commencement of other changes commonly associated with urbanization, such as alterations of water temperature and chemistry. However, the effects of land use change on ecosystems are not necessarily immediate (Harding, Benfield, Bolstad, Helfman, & Jones, 1998; Findlay & Bourdages, 2000; Löfvenhaft et al., 2004), so a lagged effect of flow regime change on fish richness is possible but could be masked by continued urbanization and other associated changes. Analyses of flow regime changes using coarser time scales would not identify event

flow acceleration alterations, especially at very low UP. The potential for early event-scale flow regime changes, as well as the potential for lagged biotic responses, should be considered as part of future analyses of declining fish richness in urbanizing watersheds. By implication, to adequately understand these potential effects, data collection is needed well in advance of large-scale urbanization; likely before 10% UP is attained.

For the recent database analyses, the addition of UP and maximum acceleration to models with total length each improved the explained variation. It must be noted that the range of flow acceleration was greatly diminished at the fish observation sites during the interval for the most recent records (see Appendix C). This reduced range of flow acceleration reflects the fact that urbanization had already dramatically altered flow regimes at the fish observation sites by the year 2000, leaving none of the sites with unaltered, low flow acceleration conditions. The reduced span raises questions about the adequacy of studies substituting space for time to assess hydrologic change and associations with fish richness in a region with decades of intense urbanization. Space will not substitute for time unless the historic range of hydrologic variables and fish richness is represented across space. The recent database had reduced power (17 vs. 49 records), so results need to be corroborated with further research in regions with comprehensive fish records and a range of UP.

Potential causal agents that could explain decreased fish diversity with increased flow acceleration are numerous and include direct effects, such as displacement of young or disruption and stress for biota when flows change at unnatural and unpredictable rates. Direct effects can be conceptualized as introducing an element of surprise for biota that are adapted for responding to subtle changes in flow as the effects of rain events move through natural watersheds. Additional research could be undertaken on the existing database to examine associations of fish species characteristics (such as body shape or reproductive habits) to develop potential hypotheses for causal relationships of flow acceleration with fish richness. Laboratory experiments, such as flume studies on fish exposed to a range of flow accelerations, may also assist in determining the importance of direct effects of event-scale hydrologic changes on fish health.

A second group of potential causal agents relate to indirect effects resulting from increased power associated with accelerating flows, including sediment transport, substrate displacement, bank erosion, etc. Interestingly, variables for stream power per unit length of channel, standardized for watershed size, were not dominant in the PCA. Additional research would be needed to assess whether this unit of estimated stream power is suitable to compare power across very different watersheds and the potential role of indirect causal agents related to event-scale flow accelerations.

The analysis did not have sufficient power to fully explore potential interactions of independent variables with respect to associations with fish richness. For example, it would be interesting to research interactions of flow acceleration with UP or watershed size to assess whether the fish richness relationship with flow acceleration differs at low levels of UP (e.g., less than 10%) or watershed area. Event flow acceleration is higher in wetter years (Trudeau & Richardson, 2015); it would also be interesting to research whether rainfall contributes to inter-annual variation in fish richness in urbanized watersheds.

If event flow acceleration negatively impacts fish communities, research would be warranted into flow accelerations during stormwater management facility operations. Objectives for stormwater volume and runoff controls (e.g., Walsh et al., 2016) would need to be refined to prevent flow accelerations from exceeding tolerable rates for resident biota. To our knowledge, flow accelerations from urban stormwater facilities have not been documented.

4.1 | Fish data bias, limitations, and issues

The myriad problems with the historical fish dataset are worth identifying because, unless an additional source of scientific data is located, these are the only data available with which to assess changes in fish richness since the mid-20th century for the study area. Additional historic fish data sources in other regions of the Great Lakes Basin or St Lawrence River may assist in supplementing scientific understanding. Single trophic level analyses of ecosystems are common (Petchey et al., 2004) yet are recognized to be inadequate to fully characterize biotic communities. Accessible benthic invertebrate datasets would also assist in understanding biotic trends in the study region.

Bias was potentially introduced into the fish dataset as a consequence of four factors: (a) advances in taxonomy, (b) varying sampling intent, (c) methodological sampling differences, and (d) level of sampling effort. Each of these factors is briefly described in the following; however, there was insufficient information to parse the effects of these potential sources of bias:

1. Taxonomic advances increased species richness temporally in the database; specifically, species were added as the science of species differentiation advanced.

2. The intent of fish surveys evolved from locating indicator species (in the 1940s and 1950s) to comprehensive surveys including estimation of abundance. The Ontario Department of Planning and Development (ODPD) surveys in the 1940s and 1950s were undertaken for indicator species using a Bayesian sampling approach based on observed stream conditions (H. Regier, pers. comm., 2014). A focus on specific fish species is evident from a field note in the 1946 ODPD survey, quantifying the number of fish present as "zero," but with the note "Intermittent pools with several small fish observed" (Lotus Data File, G.A. Wichert, pers. comm., 2011). By contrast, the OSAP Multiple Pass Survey is intended to identify all species at a site (Stanfield, 2013). Similarly, the Steedman study (Steedman, 1987) objective was to fully characterize fish communities. One objective of Wichert's research (1995) was to develop a species association tolerance index of water quality. Sampling sites were selected upstream and downstream of known pollution sources, introducing additional variation into the dataset. As wastewater treatment plant effluent releases ceased, fish community richness improved, potentially masking biodiversity trends associated with urbanization (Wichert, 1994).

3. The ODPD studies were conducted with seine nets (e.g., ODPD, 1956), whereas more recent surveys primarily use electrofishing equipment (e.g., TRCA, 2009). Seine nets are not as efficient as electrofishing techniques because they cannot be effectively deployed in areas with branches, rocky or very soft substrate, or in high velocity currents (Syracuse Research Corporation, 2001). Electrofishing is most effective in shallow streams (Syracuse Research Corporation, 2001). Steedman (1987) used both electrofishing and seine netting, which may account, in part, for a relatively high species diversity in the 1984/85 records. Species favoring deeper watercourses may be under-represented in the database due to the methodological bias towards wadable streams.

4. Effort records are not available for some studies (e.g., ODPD surveys) and units of sampling effort differ among others (e.g., Steedman, 1987; Martin, 1984). Intensive efforts may be employed to confirm the presence of endangered or rare species (e.g., TRCA, 2009). As a consequence, there may be bias towards fish communities occupying endangered species' habitats. Also, the apparent occurrence of threatened and endangered species in records subsequent to their designation may reflect increased efforts to locate them. For example, several redhorse species are recorded in the recent database but not in the long-term database.

In addition to these factors, error could have been introduced into the fish database through inappropriate grouping of fish sampling sites, discrepancies in site locations resulting from reference systems (e.g., from NAD27 to NAD83) or estimates from paper maps, and inconsistent site identification codes and site descriptions.

5 | CONCLUSIONS AND RECOMMENDATIONS

The relationships of flow acceleration and skew in instantaneous runoff with fish richness warrant further investigation because, considering the fish record limitations, the analyses of the database were not expected to have sufficient statistical power to detect any trends other than the most blatant signals. In addition to examination of associations in other watersheds, research is warranted into potential causal agents contributing to reduced fish richness associated with increasing event flow accelerations.

As a precautionary approach, flow acceleration of stormwater effluents from various types of facilities should be researched. In addition to volume and frequency controls for event flows, flow acceleration may become an important design consideration for protection of aquatic biota if further research confirms the associations identified in this study.

Historic fish data are difficult to obtain and pose analytical challenges due to bias. Use of only recent records may not present the same opportunities for analyses of flow with fish richness unless sufficient range in flow conditions is present in the database. Therefore, studies substituting space for time to compensate for a lack of historic data must ensure the range of flows within the study space that mimic the historic flow ranges, especially the lower flow and flow acceleration rates found in more natural watersheds.

Long-term fish study sites should be aligned with flow sites to the extent possible and other transdisciplinary considerations kept in mind. For example, data to assess trends in rainfall, geomorphology, water quality, water temperature, sediment transport, and habitat features would ideally be available at each fish observation site. Flow data should be collected on fine temporal scales (e.g., 15-min increments) such that event flow accelerations and instantaneous runoff can be estimated. Given the problems with historic fish data bias, data collection may need to be undertaken in watersheds proactively, as they transition from very low urban cover.

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APPENDIX

### Appendix A. Maps of watersheds in the study

Figures A1, A2, and A3 indicate the 22 watersheds in the study. See Table 1 for numbers corresponding to fish observation sites indicated on the maps.
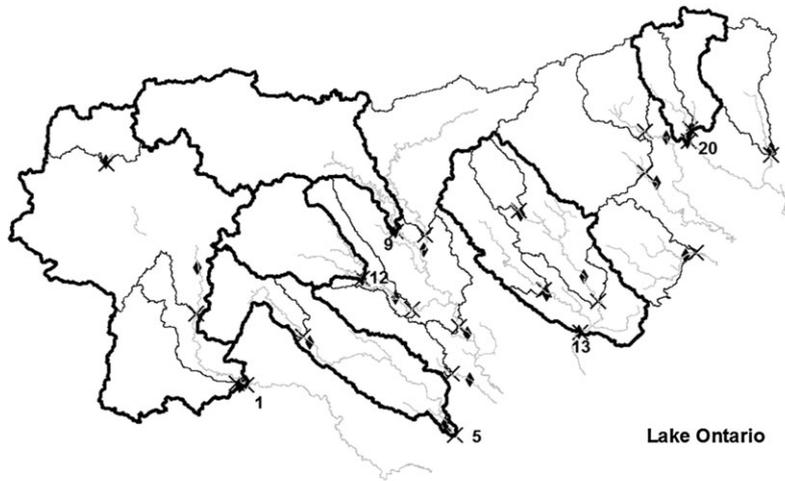
FIGURE A1 Six of the study watersheds. Six study watersheds upstream of fish observation sites are indicated by dark lines: 1. Credit mainstem, 5. Etobicoke mainstem, 9. Humber Upper mainstem, 12. Humber West branch, 13. Don mainstem, 20. Duffins West branch. Fish sites are indicated by crosses and paired Water Survey of Canada hydrologic stations by diamonds
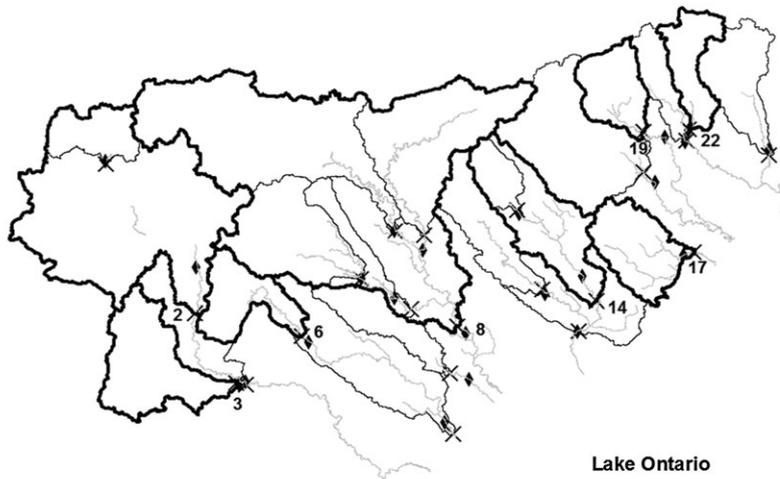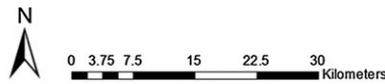

FIGURE A2 Eight of the study watersheds. Eight study watersheds upstream of fish observation sites are indicated by dark lines: 2. Credit midmainstem, 3. Credit West branch, 6. Etobicoke upper mainstem, 8. Humber mainstem, 14. Don East branch, 17. Highland mainstem, 19. Rouge East branch, 22. Duffins upper West branch. Fish sites are indicated by crosses and paired Water Survey of Canada hydrologic stations by diamonds

FIGURE A3 Eight study watersheds. Eight study watersheds upstream of fish observation sites are indicated by dark lines: 4. Credit upper mainstem, 7. Mimico mainstem, 11. Humber West branch, 10. Humber East branch, 15. Don mid-mainstem, 16. Don upper mainstem, 18. Rouge mainstem, 21. Duffins mainstem. Fish sites are indicated by crosses and paired Water Survey of Canada hydrologic stations by diamonds
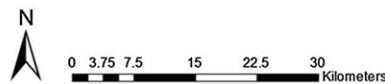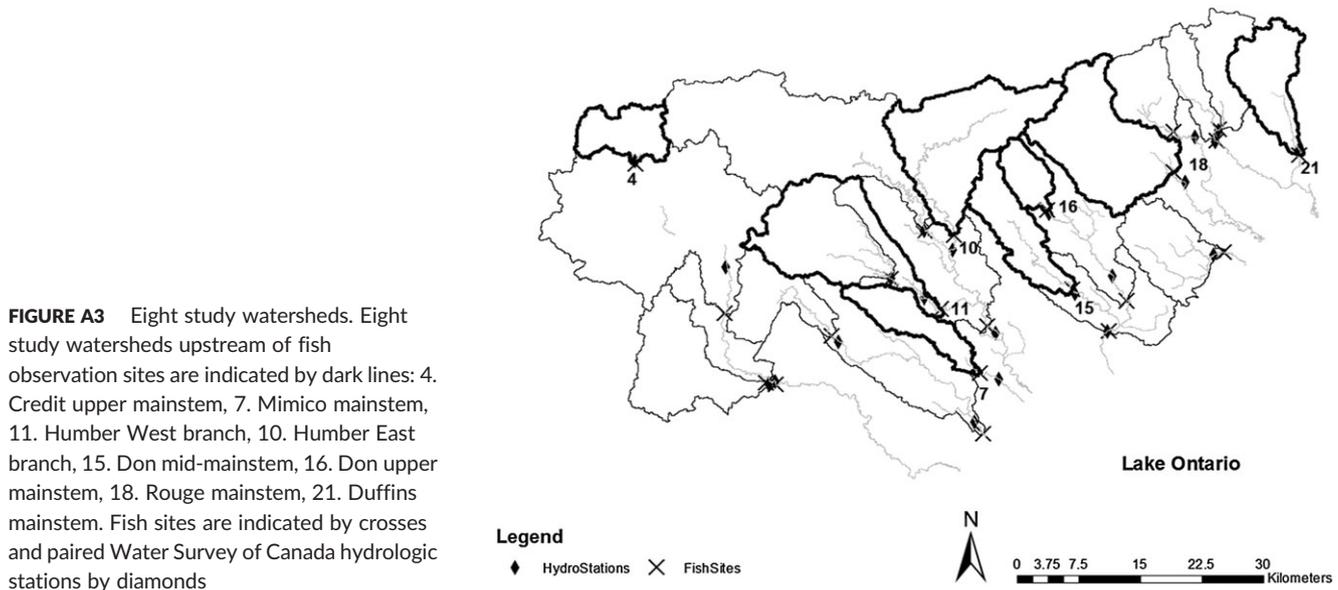
Appendix B. Fish species in long-term and recent databases

Species codes in each of the three datasets for all sites and all fish observation years available are summarized in Table B.1. In the *All records* dataset, species that are not in either of the other two datasets were present in the record after 1959 but not present in records post-1999 for the grouped sites database. These species, and others, may have been present at other sites in one or more of the eight watersheds within the study period. Due to data limitations, some species in this table were not included in the databases analyzed because the year(s) they were recorded did not have corresponding UP estimates and/or hydrologic data.

For species names, refer to the OMNR *Manual of Instructions: Nearshore Community Index Netting (NSCIN) Report* (Appendix A; Stirling, 1999). Available at URL: https://dr6j45jk9xcmk.cloudfront.net/documents/2664/226869.pdf.

TABLE B.1 Species codes in each of three databases for all available years with fish records

All records (1940s/1950s to 2010) species codes	Long-term (1940s/1950s to 2010) species codes	Recent records (2000 to 2010) species codes
S10 *Petromyzontidae*—lampreys		S10
S11 *Petromyzontidae* —lampreys; *Lethenteron appendix* (American brook lamprey)	S11	S11
S12 *Petromyzontidae*—lampreys; *Ichthyomyzon fossor* (Northern brook lamprey)		S12
S13 *Petromyzontidae*—lampreys; *Ichthyomyzon unicuspis* (silver lamprey)		
S14 *Petromyzontidae*—lampreys; *Petromyzon marinus* (sea lamprey)	S14	S14
S15 *Petromyzontidae*—lampreys; *Jchthyomyzonsp.*	S15	S15
S61 *Clupeidae*—herrings; *Alosa pseudoharengus* (alewife)	S61	S61
S63 *Clupeidae*—herrings; *Dorosoma cepedianum* (gizzard shad)		S63
S70 *Salmonidae*—salmon and trout		S70
S73 *Salmonidae*—salmon and trout; *Oncorhynchus kisutch* (coho salmon)		S73
S75 *Salmonidae*—salmon and trout; *Oncorhynchus tshawytscha* (chinook salmon)		S75
S76 *Salmonidae*—salmon and trout; *Oncorhynchus mykiss* (rainbow trout (steelhead))	S76	S76
S77 *Salmonidae*—salmon and trout; *Salmo salar* (Atlantic salmon (ouananiche))	S77	S77
S78 *Salmonidae*—salmon and trout; *Salmo trutta* (brown trout)	S78	S78
S80 *Salmonidae*—salmon and trout; *Salvelinus fontinalis* (brook (speckled) trout)	S80	S80
S103 *Coregoninae*—whitefish subfamily; *Coregonus sp.* (chub) (Cisco species other than *C. artedi*)		S103
S131 *Esocidae*—pikes; *Esox lucius* (Northern pike)	S131	S131
S141 *Umbridae*—mudminnows; *Umbra limi* (central mudminnow)	S141	S141
S160 *Catostomidae*—suckers		S160
S163 *Catostomidae*—suckers; *Catostomus commersonii* (white sucker)	S163	S163

(Continues)

TABLE B.1 (Continued)

All records (1940s/1950s to 2010) species codes	Long-term (1940s/1950s to 2010) species codes	Recent records (2000 to 2010) species codes
S165 *Catostomidae*—suckers; *Hypentelium nigricans* (Northern hog sucker)	S165	S165
S168 *Catostomidae*—suckers; *Moxostoma anisurum* (silver redhorse)		
S171 *Catostomidae*—suckers; *Moxostoma macrolepidotum* (shorthead redhorse)		S171
S172 *Catostomidae*—suckers; *Moxostoma valenciennesi* (greater redhorse)		S172
S176 *Catostomidae*—suckers;*Catostomus sp.*		S176
S177 *Catostomidae*—suckers;*Moxostoma sp.*	S177	
S180 *Cyprinidae*—carps and minnows		S180
S181 *Cyprinidae*—carps and minnows; *Carassius auratus* (goldfish)		S181
S182 *Cyprinidae*—carps and minnows; *Chrosomus eos* (Northern redbelly dace)	S182	S182
S183 *Cyprinidae*—carps and minnows; *Chrosomus neogaeus* (finescale dace)	S183	S183
S184 *Cyprinidae*—carps and minnows; *Clinostomus elongates* (redside dace)	S184	S184
S185 *Cyprinidae*—carps and minnows; *Couesius plumbeus* (lake chub)	S185	S185
S186 *Cyprinidae*—carps and minnows; *Cyprinus carpio* (common carp)	S186	S186
S189 *Cyprinidae*—carps and minnows; *Hybognathus hankinsoni* (brassy minnow)		S189
S190 *Cyprinidae*—carps and minnows; *Hybognathus regius* (Eastern silvery minnow)	S190	
S192 *Cyprinidae*—carps and minnows; *Nocomis biguttatus* (hornyhead chub)	S192	S192
S193 *Cyprinidae*—carps and minnows; *Nocomis micropogon* (river chub)	S193	S193
S194 *Cyprinidae*—carps and minnows; *Notemigonus crysoleucas* (golden shiner)	S194	S194
S195 *Cyprinidae*—carps and minnows; *Notropis anogenus* (pugnose shiner)		
S196 *Cyprinidae*—carps and minnows; *Notropis atherinoides* (emerald shiner)	S196	S196
S197 *Cyprinidae*—carps and minnows; *Notropis bifrenatus* (bridle shiner)		S197
S198 *Cyprinidae*—carps and minnows; *Luxilus cornutus* (common shiner)	S198	S198
S199 *Cyprinidae*—carps and minnows; *Notropis heterodon* (blackchin shiner)		
S200 *Cyprinidae*—carps and minnows; *Notropis heterolepis* (blacknose shiner)	S200	S200
S201 *Cyprinidae*—carps and minnows; *Notropis hudsonius* (spottail shiner)	S201	S201
S202 *Cyprinidae*—carps and minnows; *Notropis rubellus* (rosyface shiner)	S202	S202
S203 *Cyprinidae*—carps and minnows; *Cyprinella spiloptera* (spotfin shiner)	S203	S203
S204 *Cyprinidae*—carps and minnows; *Notropis stramineus* (sand shiner)		S204
S206 *Cyprinidae*—carps and minnows; *Notropis volucellus* (mimic shiner)	S206	S206
S208 *Cyprinidae*—carps and minnows; *Pimephales notatus* (bluntnose minnow)	S208	S208
S209 *Cyprinidae*—carps and minnows; *Pimephales promelas* (fathead minnow)	S209	S209
S210 *Cyprinidae*—carps and minnows; *Rhinichthys atratulus* (Eastern blacknose dace)	S210	S210
S211 *Cyprinidae*—carps and minnows; *Rhinichthys cataractae* (longnose dace)	S211	S211
S212 *Cyprinidae*—carps and minnows; *Semotilus atromaculatus* (creek chub)	S212	S212
S213 *Cyprinidae*—carps and minnows; *Semotilus corporalis* (fallfish)		S213
S214 *Cyprinidae*—carps and minnows; *Margariscus nachtriebi* (northern pearl dace)	S214	S214
S216 *Cyprinidae*—carps and minnows; *Campostoma anomalum* (central stoneroller)		S216
S217 *Cyprinidae*—carps and minnows; *Luxilus chrysocephalus* (striped shiner)		S217
S219 *Cyprinidae*—carps and minnows; *Ctenopharyngodon idella* (grass carp)		S219
S220 *Cyprinidae*—carps and minnows; *Scardinius erythrophthalmus* (rudd)		S220
S223 *Cyprinidae*—carps and minnows; *Nocomis sp.*		S223
S224 *Cyprinidae*—carps and minnows; *Notropis sp.*		S224
S225 *Cyprinidae*—carps and minnows; *Pimephales sp.*		S225
S226 *Cyprinidae*—carps and minnows; *Rhinichthys sp.*		S226
S230 *Ictaluridae*—bullhead catfishes		S230
S231 *Ictaluridae*—bullhead catfishes; *Ameiurus melas* (black bullhead)		S231
S232 *Ictaluridae*—bullhead catfishes; *Ameiurus natalis* (yellow bullhead)		S232
S233 *Ictaluridae*—bullhead catfishes; *Ameiurus nebulosus* (brown bullhead)	S233	S233
S234 *Ictaluridae*—bullhead catfishes; *Ictalurus punctatus* (channel catfish)	S234	

TABLE B.1 (Continued)

All records (1940s/1950s to 2010) species codes	Long-term (1940s/1950s to 2010) species codes	Recent records (2000 to 2010) species codes
S235 *Ictaluridae*—bullhead catfishes; *Noturus flavus* (Stonecat)	S235	S235
S236 *Ictaluridae*—bullhead catfishes; *Noturus gyrinus* (tadpole madtom)	S236	
S239 *Ictaluridae*—bullhead catfishes; *Pylodictis olivaris* (flathead catfish)		S239
S241 *Ictaluridae*—bullhead catfishes; *Ictalurus sp.*		
S242 *Ictaluridae*—bullhead catfishes; *Noturus sp.*	S242	
S251 *Anguillidae*—freshwater eels; *Anguilla rostrate* (American eel)		S251
S261 *Cyprinodontidae*—killifishes; *Fundulus diaphanous* (banded killifish)	S261	
S271 Gadidae—cods; *Lota lota* (burbot)		
S280 *Gasterosteidae*—sticklebacks		S280
S281 *Gasterosteidae*—sticklebacks; *Culaea inconstans* (brook stickleback)	S281	S281
S282 *Gasterosteidae*—sticklebacks; *Gasterosteus aculeatus* (three-spined stickleback)	S282	S282
S291 *Percopsidae*—trout-perches; *Percopsis omiscomaycus* (trout-perch)	S291	S291
S301 *Percichthyidae*—temperate basses; *Morone americana* (white perch)		
S302 *Percichthyidae*—temperate basses; *Morone chrysops* (white bass)		S302
S310 *Centrarchidae*—sunfishes		S310
S311 *Centrarchidae*—sunfishes; *Ambloplites rupestris* (rock bass)	S311	S311
S312 *Centrarchidae*—sunfishes; *Lepomis cyanellus* (green sunfish)		S312
S313 *Centrarchidae*—sunfishes; *Lepomis gibbosus* (pumpkinseed)	S313	S313
S314 *Centrarchidae*—sunfishes; *Lepomis macrochirus* (bluegill)		S314
S316 *Centrarchidae*—sunfishes; *Micropterus dolomieu* (smallmouth bass)	S316	S316
S317 *Centrarchidae*—sunfishes; *Micropterus salmoides* (largemouth bass)	S317	S317
S319 *Centrarchidae*—sunfishes; *Pomoxis nigromaculatus* (black crappie)		S319
S320 *Centrarchidae*—sunfishes; *Lepomis sp.*		S320
S330 *Percidae*—perches		S330
S331 *Percidae*—perches; *Perca flavescens* (yellow perch)	S331	S331
S334 *Percidae*—perches; *Sander vitreus* (walleye [yellow pickerel])		
S337 *Percidae*—perches; *Etheostoma caeruleum* (rainbow darter)	S337	S337
S338 *Percidae*—perches; *Etheostoma exile* (Iowa darter)	S338	S338
S339 *Percidae*—perches; *Etheostoma flabellare* (fantail darter)	S339	S339
S340 *Percidae*—perches; *Etheostoma microperca* (least darter)		
S341 *Percidae*—perches; *Etheostoma nigrum* (johnny darter)	S341	S341
S342 *Percidae*—perches; *Percina caprodes* (logperch)		S342
S344 *Percidae*—perches; *Percina maculate* (blackside darter)		S344
S345 *Percidae*—perches; *Percina shumardi* (river darter)		
S346 *Percidae*—perches; *Etheostoma olmstedi* (tessellated darter)		S346
S348 *Percidae*—perches; *Etheostoma sp.*		S348
S351 *Percidae*—perches; *Etheostoma nigrum/Etheostoma olmstedi* (johnny/tesselated darter hybrid)		S351
S366 *Gobiidae*—gobies; *Neogobius melanostomus* (round goby)		S366
S371 *Sciaenidae*—drums; *Aplodinotus grunniens* (freshwater drum)		S371
S380 *Cottidae*—sculpins		S380
S381 *Cottidae*—sculpins; *Cottus bairdii* (mottled sculpin)	S381	S381
S382 *Cottidae*—sculpins; *Cottus cognatus* (slimy sculpin)	S382	S382
S600 *Cyprinidae*—hybrids		S600
S612 *Cyprinidae*—hybrids; *Luxilus cornutus x Semotilus atromaculatus*		S612
S703 *Centrarchidae*—hybrids; *Lepomis cyanellus x Lepomis gibbosus*		S703
S921 900—unidentifiable (Sandstrom, Rawson, & Lester, 2013)		S921

Appendix C. Comparison of independent variable ranges in the long-term and recent databases

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TABLE C.1 Range of hydrologic variables in two databases: long-term database and recent database. Comparison of the span of select variables in the long-term database versus the recent database for maximum record, minimum record, and span

Dataset	Long-term fish database (49 records)			Recent records database (17 records)		
Variable	Minimum	Maximum	Comparison with recent database	Minimum	Maximum	Comment
Year	1959	2010		2000	2010	
Total fish species	0	18		2	18	
Urban percent	0.06	87.6	Very low minimum urban cover in long-term database	9.2	87.6	Minimum in recent database 150 times higher than minimum in long-term database
MF·m2	2.16e-10	1.55e-08	Ratio max: minimum ~70 versus ~5 for recent database	3.38e-09	15.53e-09	Minimum in recent database 1.6 times higher than minimum in long-term database
Skew in runoff	1.21	13.13	Ratio max: minimum 10.8 versus 9.7 for recent database	1.21	11.70	Same minimum skew
Maximum acceleration	6.78e-05	0.0603	Ratio max: minimum ~900 versus ~400 for recent database	0.00015	0.0603	Minimum in recent database 2 times higher than minimum in long-term database
Skew acceleration	3.24	35.53	Ratio max: minimum ~10 versus ~4 for recent database	4.53	17.48	
90th percentile power per length channel per square meter drainage area	0.26e-07	23.53e-07	Ratio max: minimum ~89 versus ~14 for recent database	1.68e-07	23.53e-07	Minimum in recent database 6 times higher than minimum in long-term database